Global variation in terrestrial conservation costs, conservation benefits, and unmet conservation needs

Andrew Balmford*[†], Kevin J. Gaston[‡], Simon Blyth[§], Alex James[¶], and Val Kapos[§]

*Conservation Biology Group, Department of Zoology, University of Cambridge, Downing Street, Cambridge CB2 3EJ, United Kingdom; [‡]Biodiversity and Macroecology Group, Department of Animal and Plant Sciences, University of Sheffield, Sheffield S10 2TN, United Kingdom; [§]UNEP World Conservation Monitoring Centre, 219 Huntingdon Road, Cambridge CB3 ODL, United Kingdom; and [¶]Department of Land Economy, University of Cambridge, 19 Silver Street, Cambridge CB3 9EP, United Kingdom

Communicated by Norman Myers, University of Oxford, Oxford, United Kingdom, November 14, 2002 (received for review September 20, 2002)

Our ability to identify cost-efficient priorities for conserving biological diversity is limited by the scarcity of data on conservation costs, particularly at fine scales. Here we address this issue using data for 139 terrestrial programs worldwide. We find that the annual costs of effective field-based conservation vary enormously, across seven orders of magnitude, from <$0.1 to >$1,000,000 per km^2. This variation can be closely predicted from positive associations between costs per unit area and an array of indices of local development. Corresponding measures of conservation benefit are limited but show opposing global trends, being higher in less developed parts of the world. The benefit-to-cost ratio of conservation is thus far greater in less developed regions, yet these are where the shortfall in current conservation spending is most marked. Substantially increased investment in tropical conservation is therefore urgently required if opportunities for cost-effective action are not to be missed.

The gross mismatch between the costs of effective nature conservation and current global spending (1) means that prioritization of conservation effort is essential and should be based on economic as well as biological information (2–5). Moreover, because a large portion of resources for conservation now comes from intergovernmental organizations and major private foundations, such priority setting needs to be conducted at global as well as regional and local levels. However, the scarcity of data on how conservation costs vary globally, particularly at the resolution of individual programs, means that in practice economic considerations are commonly ignored.

To tackle this problem, we collated information on the costs of 139 field-based projects from around the world. We compared variation in costs with a suite of measures of development, built a simple model capable of predicting costs elsewhere, and explored global variation in likely conservation benefits. Finally, we compared our findings on conservation costs and benefits with the current global distribution of conservation investment.

Data

Conservation Costs. We obtained data on the recurrent management costs per unit area of effective terrestrial field-based conservation programs (expressed in year-2000 U.S. dollars) from United Nations Environment Programme–World Conservation Monitoring Centre (UNEP-WCMC) surveys of protected area agencies (ref. 6; 57 sites), from correspondence with local experts (21 sites), from the published and unpublished literature (20 sites), and from the World Wide Web (41 sites). In the first two cases, correspondents were explicitly asked whether current spending was sufficient to meet conservation goals and if not, how much extra would be required. Similarly, all data from the literature and web sites referred to total costs for effective conservation and, for all but 13 of these cases, this was broken down into current spending and unmet costs. Note that all our results were qualitatively unchanged when analyses were restricted to the 57 sites surveyed by the most widely used method, the UNEP-WCMC questionnaire.

In sum, our cost dataset spanned 37 nations from all major landmasses except Antarctica, consisted mostly of reserves but also covered conservation programs in the wider landscape, and included 64 projects from less developed countries. All costs (including regional investment patterns and needs; see below) were converted to U.S. dollars by using contemporary exchange rates and then converted between years by using a U.S. gross domestic product deflator index (7).

We were unable to obtain figures for our sample of sites for other costs of conservation besides on-site management, such as opportunity costs, land purchase, transactions costs, or the costs of tackling larger-scale threats such as landscape-wide overexploitation or changes to fire or hydrological regimes. Few such data exist (1, 8), and those that do are generally from a different subset of conservation initiatives. Although this is a limitation of the present findings, we have shown elsewhere (9) that land purchase costs are reasonably closely related to annual recurrent management costs [for eight countries, national mean land purchase costs km^{-2} ran at a mean ($\pm$SE) of 48.9 ($\pm$21.2) times national mean recurrent costs km^{-2} y^{-1} (9); for an enlarged sample of 19 countries, the ratio is now 50.6 ($\pm$13.5)]. Hence, although obtaining direct data on other costs would be invaluable, we do not believe it would have a major impact on our conclusions.

Measures of Development. We quantified development using a suite of local and national measures (Table 1). Wilderness values were provided by the Australian Heritage Commission for all 2.5 $\times$ 2.5-km pixels of the earth's land surface (10). The scores integrate the distance of a site from permanent settlement, from built access routes, and from other built infrastructure, and run from 0 (fully converted) to 22 (highest wilderness) (10). Values for all pixels overlain by our sites were extracted and averaged by using ARCINFO software (ESRI, Redlands, CA). Local population densities (people·km^{-2}) were extracted in a similar way from a global surface modeled at 5′ resolution (11). Unlike these local measures, all other indices of development that we used refer to national averages. Gross national product (GNP), population, country area, and purchasing power parity (PPP) were obtained for 1999 from ref. 12, supplemented for nonreporting countries by estimates kindly provided by World Bank staff.

Conservation Benefits. There are very few data with which to compare conservation benefits across globally scattered sites; we looked here at two simple measures. First, given concerns that many of the evolutionary and ecological processes that underpin biological diversity can be maintained only in large blocks of habitat (13), one measure of likely conservation benefit is the total area that could be properly conserved for a given annual investment. This is of course simply the reciprocal of the annual cost per unit area, as derived above.

Second, turning to the conservation of species, BirdLife

Abbreviations: PPP, purchasing power parity; GNP, gross national product.

[†]To whom correspondence should be addressed. E-mail: a.balmford@zoo.cam.ac.uk.

Table 1. Matrix of Pearson correlation coefficients between our measures of development

	Population density, km$^{-2\dagger}$	Per capita GNP, \$ y^{-1}	GNP per unit area, \$ km^{-2} y^{-1}	PPP	Project area, km^2
Wilderness value	−0.52*	−0.35*	−0.47*	0.36*	0.50*
Population density, km$^{-2\dagger}$		0.01, NS	0.42*	−0.03, NS	−0.30*
Per capita GNP, \$ y^{-1}			0.84*	−0.93*	−0.46*
GNP per unit area, \$ km^{-2} y^{-1}				−0.78*	−0.51*
PPP					0.51*

df = 137 throughout; P values are adjusted by using Bonferroni correction. All variables $\log_{10}$-transformed, except: †, $\log_{10}$ (population density + 0.05)-transformed. *, $P < 0.05$; NS, not significant.

International's recent map of threatened bird species richness across 1/4° grid squares (14) provides an absolute measure of the global biodiversity loss that might be avoided by effective field efforts at each of our sites. We extracted numbers of threatened bird species for all 1/4° grid squares overlain by each of our field-based projects by using an electronic version of data in *Threatened Birds of the World* (14). A problem here is that although reserves and other conservation initiatives may be expected to target threatened species, not all those in a 1/4° grid square may be present in a site if it does not completely cover the square, hence this approach exaggerates the conservation benefits of individual projects. However, as the proportion of squares that are not fully overlapped decreases with increasing project area, this error biases us toward overestimating the relative benefits of smaller reserves in particular and so is conservative in terms of the pattern observed here (see below).

Current Conservation Investment. We examined coarse-scale variation in conservation investment in terrestrial reserves by extracting estimates of current and necessary spending for each of 10 regions of the world from James *et al.* (9). Unlike our site-based estimates, these figures included not just regional estimates of current and unmet spending on managing existing reserves (as reported by protected area agencies, ref. 6) but also (for existing category II–IV reserves in developing countries) estimates of the opportunity costs accruing to local communities, plus the survey, land purchase, and effective management costs associated with expanding the network to cover ≈15% of each region (see ref. 9 for details).

Data Analysis. Most analyses were nonparametric. Where data were analyzed parametrically, they were first $\log_{10}$-transformed to achieve approximate normality (including zeroes by adding 0.05, 0.1, and 0.001, respectively, to scores for population density, threatened bird density, and the ratio of threatened bird density to annual cost km^{-2}). In multiple regression, best models were sought by using both forward and backward stepwise procedures, with F to enter (and remove) set to 4.0. We did not fit interaction terms in any of the models. For both the final models, reported visual inspection of residual versus fitted values revealed an anomalous group of 11 sites: all of those in Chad, Mongolia, and Russia had strong negative residuals despite very low fitted costs. However, removing these points had no substantive effect on the results reported here.

Results and Discussion

Conservation Costs. Unsurprisingly, we found that the costs of field conservation vary widely. What is more striking is the scale of the variation: costs range over seven orders of magnitude, from less than \$0.1 km^{-2} y^{-1} in the Russian Arctic to over \$1,000,000 km^{-2} y^{-1} for some western European programs in which restoration is needed to recover conservation value. This variation is correlated with the extent of nearby development. (e.g., costs vs. wilderness value: $r_s = -0.55$, $n = 139$ sites, $P < 0.001$; Fig. 1*a*). For areas with high wilderness scores, such as the Gobi Desert, the Himalayas, and the Amazon, costs of effective reserves vary from <\$1–60 km^{-2} y^{-1}, but typically lie around \$20 km^{-2} y^{-1}. Costs of effective protected areas in more densely settled regions of Latin and Central America, Africa, and Asia [many of which lie in Norman Myers' and Conservation International's "hotspots" of high endemism and threat (15)] range from \$130 to >\$5,000 km^{-2} y^{-1}, with typical costs of ≈\$1,000 km^{-2} y^{-1}. In the developed world, costs differ widely, but include figures of \$5,000 to >\$40,000 km^{-2} y^{-1} for sampled U.S. nature reserves, and \$15–50,000 km^{-2} y^{-1} for U.K. reserves and agri-environment programs. For comparison (but not further analysis), *ex situ* conservation in three well-respected U.K. and U.S. zoos costs between \$6,000,000 and \$160,000,000 km^{-2} y^{-1}, reinforcing the point that, where possible, zoos could increase the cost effectiveness of their contribution to conservation by supporting field-based initiatives (16).

Conservation costs are correlated with local human population density ($r_s = 0.36$, $n = 139$, $P < 0.001$) and increase closely with economic activity, as measured by mean per capita GNP ($r_s = 0.75$, $n = 139$, $P < 0.001$; Fig. 1*b*) or the ratio of GNP to country area ($r_s = 0.80$, $n = 139$, $P < 0.001$). Dollar costs decrease with increases in the local buying power of a U.S. dollar (measured as PPP: $r_s = -0.80$, $n = 139$, $P < 0.001$). Costs per unit area also decrease with the areal extent of projects ($r_s = -0.69$, $n = 139$, $P < 0.001$), with the slope of the regression of log (annual cost, in dollars km^{-2} y^{-1}) against log (area, in km^2) being similar to a recent figure for the Cape Floristic Region (17)

ECOLOGY

Table 2. Building multiple regression models of variation in annual management cost and benefit-to-cost ratio of field conservation projects

		Independent variables (coefficient, t)			
Dependent variable	Intercept	GNP per unit area, \$ km^{-2} y^{-1}	PPP	Project area, km^2	Overall r^2
Cost, \$ km^{-2} y^{-1}	1.61***	0.57, 8.13***	−0.70, 2.34*	−0.46, −9.12***	0.81***
Ratio of number of threatened bird species per 1/4° grid to cost, \$ km^{-2} y$^{-1\dagger}$	0.35, NS	−0.45, 4.74***	3.84, 8.50***	0.22, 2.46**	0.76***

$n = 139$ throughout. All variables $\log_{10}$-transformed except: †, $\log_{10}$ (ratio of number of threatened bird species per 1/4° grid to cost) + 0.001-transformed. *, $P < 0.05$; **, $P < 0.01$; ***, $P < 0.001$; NS, not significant.

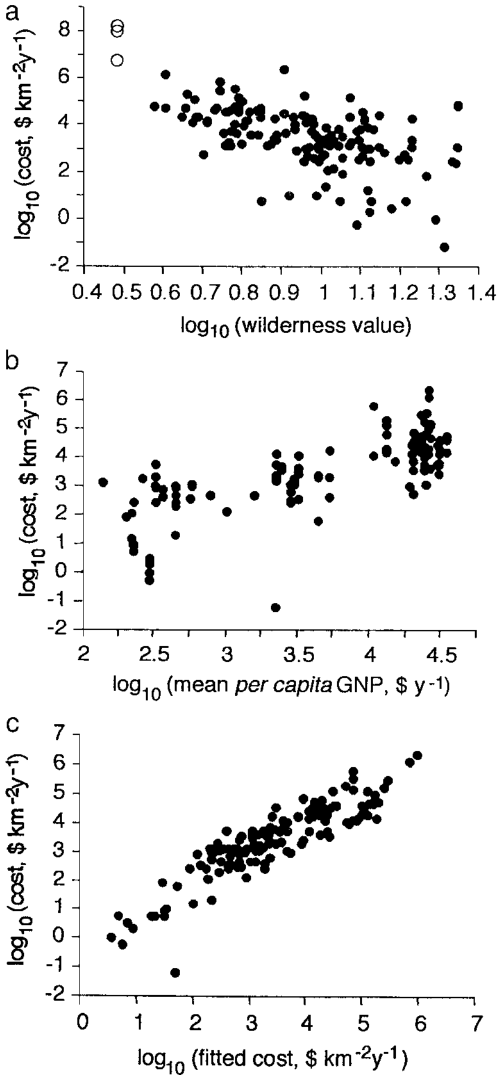

Fig. 1. Variation in the annual management cost of conservation projects; note axes are $\log_{10}$-transformed. (*a*) Annual cost km^{-2} vs. wilderness value; for comparison only, the open symbols are well-respected U.K. and U.S. zoos; these are not included in any analysis. (*b*) Annual cost km^{-2} vs. mean per capita GNP. (*c*) Observed vs. fitted values from the multiple regression model of annual cost given in Table 2.

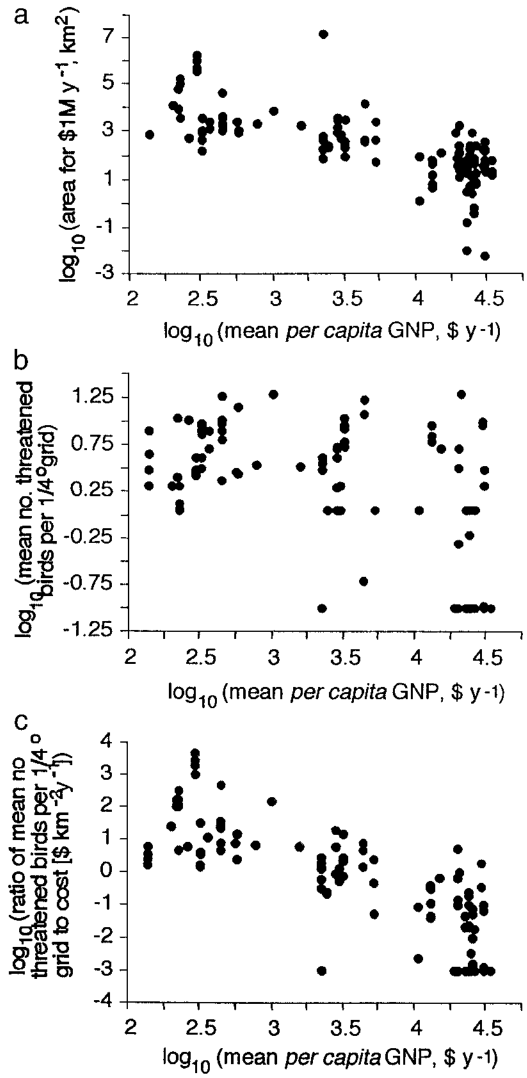

Fig. 2. Variation in the estimated benefits of field-based conservation projects. (*a*) Area that could be effectively conserved for \$1,000,000 y^{-1} vs. mean per capita GNP. (*b*) Mean number of threatened bird species per 1/4° grid vs. mean per capita GNP. (*c*) Ratio of threatened bird density to annual cost km^{-2} vs. mean per capita GNP.

[-0.85 (SE $\pm$ 0.07), compared to -0.70), perhaps reflecting consistent economies of scale.

Because many of these variables are correlated with one another (Table 1), dissecting their independent relationships with conservation cost requires multiple regression analysis. Forward and backward stepwise procedures identified the same model, with log (GNP per unit area), log (PPP), and log (area) together predicting >80% of the variance in log (annual cost km^{-2}) (Table 2; Fig. 1*c*). Hence effective field-based conservation efforts are cheaper when conducted in less developed regions with low cost structures, and where they cover large areas.

Conservation Benefits. Although being able to predict the costs of effective conservation is useful, the significance of this result for prioritization also depends on the distribution of conservation benefits in relation to development. The first of our benefit measures, the total area that could be effectively conserved for a fixed annual spend, is of course the reciprocal of annual cost per unit area and as such increases with wilderness value, PPP, and a project's areal extent and decreases with local population density, per capita GNP (Fig. 2*a*), and GNP per unit area.

Turning to our second measure of conservation benefit, we found that the mean number of threatened bird species occurring in the 1/4° grids overlapped by a project is generally lower in more developed areas (although the strength of correlations is quite variable). Threatened bird density increases with wilderness value ($r_s = 0.25$, $n = 139$, $P < 0.01$), PPP ($r_s = 0.64$, $n = 139$, $P < 0.001$) and project area ($r_s = 0.24$, $n = 139$, $P < 0.01$); decreases with per capita GNP ($r_s = -0.57$, $n = 139$, $P < 0.001$; Fig. 2*b*) and GNP per unit area ($r_s = -0.42$, $n = 139$, $P < 0.001$); and is independent of human population density ($r_s = 0.02$, $n = 139$, not significant).

These results appear counterintuitive in the light of national and continental analyses linking high rates of threat and extinction with dense human settlement (18–22). Moreover, our global findings are generally reversed when looking just across sites in developing countries (threatened bird density vs. wilderness value: $r_s = -0.22$, $P < 0.1$; vs. PPP: $r_s = -0.02$, not significant; vs. project area: $r_s = -0.40$, $P < 0.01$; vs. per capita GNP: $r_s = 0.32$, $P < 0.05$; vs. GNP per unit area: $r_s = 0.51$, $P < 0.001$; vs.

human population density: $r_s = 0.43$, $P < 0.001$; $n = 64$ sites throughout). However, in the global comparison, which is most directly relevant to global prioritization, these finer-scale patterns are swamped by gross latitudinal differences in development and biogeography, with development most intense in temperate regions, where its impact in terms of species loss is generally buffered by low species richness, large species range size, and high population densities of individual species (23, 24).

Benefit-to-Cost Ratios. With markedly lower conservation costs and generally greater conservation benefits, field programs typically have far higher benefit-to-cost ratios in less developed parts of the world. We found that the ratio of threatened bird density to annual cost per unit area increases with wilderness value ($r_s = 0.43$, $n = 139$, $P < 0.01$), PPP ($r_s = 0.86$, $n = 139$, $P < 0.001$), and project area ($r_s = 0.55$, $n = 139$, $P < 0.001$), and decreases with human population density ($r_s = -0.21$, $n = 139$, NS), per capita GNP ($r_s = -0.82$, $n = 139$, $P < 0.001$; Fig. 2*c*), and GNP per unit area ($r_s = -0.67$, $n = 139$, $P < 0.001$).

Multiple regression analysis identified the same predictors of benefit-to-cost ratio as were identified for costs alone (Table 2). This indicates that conservation projects yield the greatest benefit for a given investment when they are located in regions where GNP per unit area is low, cost structures are low, and reserves or other initiatives can cover large areas.

Current Conservation Investment. Ironically, however, these are precisely the regions where current conservation spend is lowest, and where unmet conservation needs are greatest. Of an estimated total of $6 billion spent each year on managing protected areas, for example, <12% is spent in less developed countries, where most biodiversity occurs (9). This is not simply a reflection of lower overall costs in these areas: although our sample is inevitably biased toward high profile well-funded projects, many are nevertheless underfunded, with the shortfall greatest in least developed places (percentage of total management cost that is currently met vs. wilderness value: $r_s = -0.18$, $P < 0.05$; vs. PPP: $r_s = -0.40$, $P < 0.001$; vs. population density: $r_s = 0.11$, not significant; vs. per capita GNP: $r_s = 0.48$, $P < 0.001$; vs. GNP per unit area: $r_s = 0.38$, $P < 0.001$; $n = 126$ throughout).

Moving to a coarse scale, we found that this pattern of greater shortfall in less developed areas is even stronger when other costs, associated with network expansion, are considered and compared across entire regions (9). The extent to which the overall cost of effective reserve networks is currently met is very low indeed in less developed regions (% overall costs met vs. regional mean per capita GNP: $r_s = 0.79$, $n = 10$ regions, $P < 0.05$; Fig. 3).

Conclusion

We draw two main conclusions from our work. First, the costs of effective field-based terrestrial conservation vary enormously, and interestingly apparently more than do the likely benefits. Although measuring benefit is nonetheless extremely valuable, our results highlight the need for more thorough documentation of conservation costs, across a broader range of projects, and encompassing aspects of cost (such as opportunity and transactions costs and wider landscape costs) that we were unable to consider here.

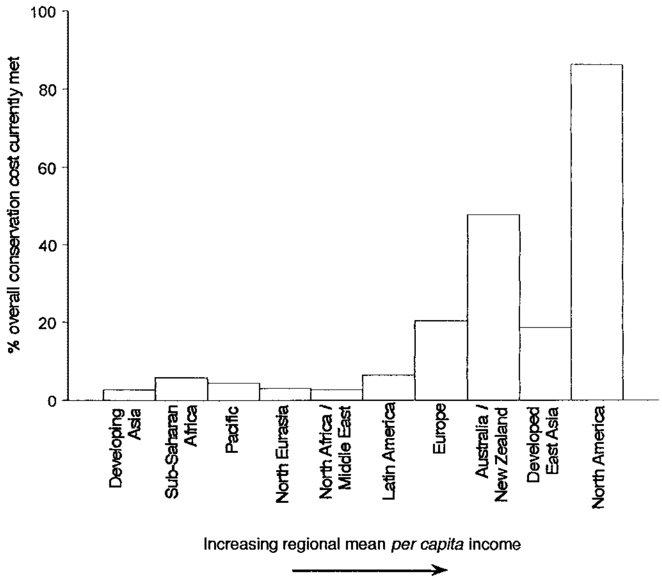

Fig. 3. Regional variation in the percentage of the overall cost of effective reserve networks that are met. These figures refer to the estimated overall costs of expanded networks (from ref. 9), but the positive correlation with mean regional GNP holds also for the percentage of existing reserve management costs that is currently met ($r_s = 0.72$, $n = 10$, $P < 0.05$).

Second, our findings underline the pressing need for much greater conservation investment by the global conservation community in places where the costs of effective conservation are relatively low and the benefits generally high. Priority areas for increased investment include developing country "hotspots" (15) where, despite high threat, costs are still generally lower than in Europe and North America. But our results also provide fresh quantitative support for calls (15, 25, 26) for high priority to be afforded to some more isolated areas as well, where conservation benefits can be substantial and conservation costs are often extremely low. Local support and involvement are of course morally, practically, and politically essential for successful conservation wherever it takes place, but we believe that addressing spending shortfalls of the magnitude illustrated in Fig. 3 will almost always require substantial north–south transfer of resources as well (27).

Most importantly, our plots of costs and benefits against development provide a sobering reminder that we cannot afford to wait (see also ref. 28); unless we protect relatively intact ecosystems while we can, not only will we be able to conserve less, but the costs of doing so will have greatly increased.

We are very grateful to Michael Green, Jerry Harrison, and Christoph Zockler at UNEP World Conservation Monitoring Centre and anonymous colleagues around the world for providing cost data; to John Dixon and Katie Bolt at the World Bank for providing supplementary economic information; to Mark Balman, Melanie Heath, and Alison Stattersfield at BirdLife International for providing data on threatened birds; and to Paul Ferraro, Rhys Green, Melanie Heath, Jos Moore, and Hugh Possingham for commenting on an earlier version of this paper.

1. James, A. N., Gaston, K. J. & Balmford, A. (1999) *Nature* **401,** 323–324.
2. Margules, C. R. & Pressey, R. L. (2000) *Nature* **405,** 243–253.
3. Ando, A., Camm, J., Polasky, S. & Solow, A. (1998) *Science* **279,** 2126–2128.
4. Williams, P. H. (1998) in *Conservation in a Changing World*, eds. Mace, G. M., Balmford, A. & Ginsberg, J. R. (Cambridge Univ. Press, Cambridge, U.K.), pp. 211–249.
5. Balmford, A., Gaston, K. J., Rodrigues, A. S. L. & James, A. (2000) *Conserv. Biol.* **14,** 597–605.
6. James, A. N., Green, M. J. B. & Paine, J. R. (1999) *A Global Review of Protected Area Budgets and Staff* (World Conservation Monitoring Centre, Cambridge, U.K.).
7. International Monetary Fund (2001) *International Financial Statistics* (International Monetary Fund, Washington, DC).
8. Ferraro, P. J. (2001) *The Local Costs of Establishing Protected Areas in Low-Income Nations: Ranomafana National Park, Madagascar* (Environmental Policy Working Paper 2001-006, Georgia State University, Atlanta).
9. James, A. N., Gaston, K. J. & Balmford, A. (2001) *BioScience* **51,** 43–52.

10. Lesslie, R. & Maslen, M. (1995) *National Wilderness Inventory Handbook of Procedures, Content and Usage* (Commonwealth Government Printer, Canberra, Australia), 2nd Ed.
11. Tobler, W., Deichmann, U., Gottsegen, J. & Maloy, K. (1995) *The Global Demography Project* (National Center for Geographic Information and Analysis Technical Report TR-95-6, University of California, Santa Barbara, CA).
12. World Bank (2001) *World Development Report 2000/2001* (Oxford Univ. Press, Oxford, U.K.).
13. Mace, G. M., Balmford, A. & Ginsberg, J. R., eds. (1998) *Conservation in a Changing World* (Cambridge Univ. Press, Cambridge, U.K.).
14. BirdLife International. (2000) *Threatened Birds of the World* (Lynx Edicions and BirdLife International, Barcelona and Cambridge, U.K.).
15. Myers, N., Mittermeier, R. A., Mittermeier, C. G., da Fonseca, G. A. B. & Kent, J. (2000) *Nature* **403,** 853–858.
16. Balmford, A., Leader-Williams, N. & Green, M. J. B. (1995) *Biodiv. Conserv.* **4,** 595–607.
17. Frazee, S. R., Cowling, R. M., Pressey, R. L., Turpie, J. K. & Lindberg, N. (2003) *Biol. Conserv.*, in press.
18. Abbitt, R. J. F., Scott, J. M. & Wilcove, D. S. (2000) *Biol. Conserv.* **96,** 169–175.
19. Rivard, D. H., Poitevin, J., Plasse, D., Carleton, M. & Currie, D. J. (2000) *Conserv. Biol.* **14,** 1099–1109.
20. Woodroffe, R. (2000) *Anim. Conserv.* **3,** 165–173.
21. Balmford, A., Moore, J. L., Brooks, T., Burgess, N., Hansen, L. A., Williams, P. H. & Rahbek, C. (2001) *Science* **291,** 2616–2619.
22. Brashares, J. S., Arcese, P. & Sam, M. K. (2001) *Proc. R. Soc. London Ser. B* **268,** 2473–2478.
23. Gaston, K. J. & Blackburn, T. M. (1996) *Proc. R. Soc. London Ser. B* **263,** 235–240.
24. Gaston, K. J. & Blackburn, T. M. (2000) *Pattern and Process in Macroecology* (Blackwell Science, Oxford).
25. Mittermeier, R. A., Myers, N., Thomsen, J. B., da Fonseca, G. A. B. & Olivieri, S. (1998) *Conserv. Biol.* **12,** 516–520.
26. Olson, D. & Dinerstein, E. (1998) *Conserv. Biol.* **12,** 502–515.
27. Balmford, A. & Whitten, T. (2003) *Oryx*, in press.
28. Chen, L. Y. (2001) *Nat. Areas J.* **21,** 197–203.